

Water quality guidelines for the Great Barrier Reef World Heritage Area: a basis for development and preliminary values [☆]

Andrew Moss ^{a,*}, Jon Brodie ^b, Miles Furnas ^c

^a *Queensland Environmental Protection Agency, Brisbane, Australia*

^b *Australian Centre for Tropical Freshwater Research, James Cook University, Townsville, Australia*

^c *Australian Institute of Marine Science, Townsville, Australia*

Abstract

The Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC Guidelines) provide default national guideline values for a wide range of indicators of relevance to the protection of the ecological condition of natural waters. However, the ANZECC Guidelines also place a strong emphasis on the need to develop more locally relevant guidelines. Using a structured framework, this paper explores indicators and regional data sets that can be used to develop more locally relevant guidelines for the Great Barrier Reef World Heritage Area (GBRWhA). The paper focuses on the water quality impacts of adjacent catchments on the GBRWhA with the key stressors addressed being nutrients, sediments and agricultural chemicals. Indicators relevant to these stressors are discussed including both physico-chemical pressure indicators and biological condition indicators. Where adequate data sets are available, guideline values are proposed. Generally, data were much more readily available for physico-chemical pressure indicators than for biological condition indicators. Specifically, guideline values are proposed for the major nutrients nitrogen (N) and phosphorus (P) and for chlorophyll-*a*. More limited guidelines are proposed for sediment related indicators. For most agricultural chemicals, the ANZECC Guidelines are likely to remain the default of choice for some time but it is noted that there is data in the literature that could be used to develop more locally relevant guidelines.

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1. Introduction

In Australia, the National Water Quality Management Strategy (ARMCANZ and ANZECC, 1994) provides a national framework for management of water quality. This framework is cyclic and involves determining desired values and objectives (or targets), implementing management strategies to achieve these objectives, and a monitoring or audit process providing a feedback loop to assess the effectiveness of the objectives and management strategies.

Integral to this process is the establishment of appropriate water quality guidelines. These are defined as numerical limits or narrative statements designed to support and maintain a designated water use, and provide the technical basis for determining quantitative management objectives or targets. A second important use of guidelines is as benchmarks for assessment of system condition. In this context, guideline exceedance is regarded as a trigger to further investigation rather than failure of a mandatory standard, hence guidelines are sometimes termed trigger values. Guideline values are based on the best technical information available at the time.

National water quality guidelines for Australia have been developed and upgraded over the last 20 years (Hart, 1982; ANZECC, 1992; ANZECC, 2000). The most recent guidelines are built on principals of

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* Corresponding author.

E-mail address: andrew.moss@epa.qld.gov.au (A. Moss).

ecosystem-based, issue-based and risk-based management (Hart et al., 1999).

The current Australian water quality guidelines (ANZECC, 2000) provide default guidelines for a wide range of indicators. However, a key philosophy of the national guidelines is that, wherever possible, regional or local guidelines should be developed that are better tailored to local conditions and approaches to developing local guidelines are provided.

The purpose of this paper is to describe the initial phases in development of appropriate regional water quality guidelines for the Great Barrier Reef World Heritage Area (GBRWHA) based on the philosophies and methodologies contained in the ANZECC, 2000 Guidelines.

Initial attempts to develop water quality guidelines for the Great Barrier Reef (GBR) and other coral reef systems were carried out by Hawker and Connell

(1989) and Bell et al. (1989). These were based on limited information available at the time. These earlier values are compared with the values derived here.

2. Great Barrier Reef World Heritage Area

The Great Barrier Reef World Heritage Area (GBRWHA) includes the estuarine, coastal and offshore areas adjacent to the NE coast of Australia between Cape York (9°S) and Bundaberg (24°S) (Fig. 1). The GBRWHA and its management are described in more detail elsewhere (Brodie, 2002, 2003; Furnas, 2003).

The main components of the system are shown in Fig. 2. These include estuaries and shallow inshore areas with extensive areas of mangroves and seagrass, inshore reefs and islands with fringing reefs (the inner-shelf reefs), the reef lagoon and the main barrier reef (outer shelf reefs).

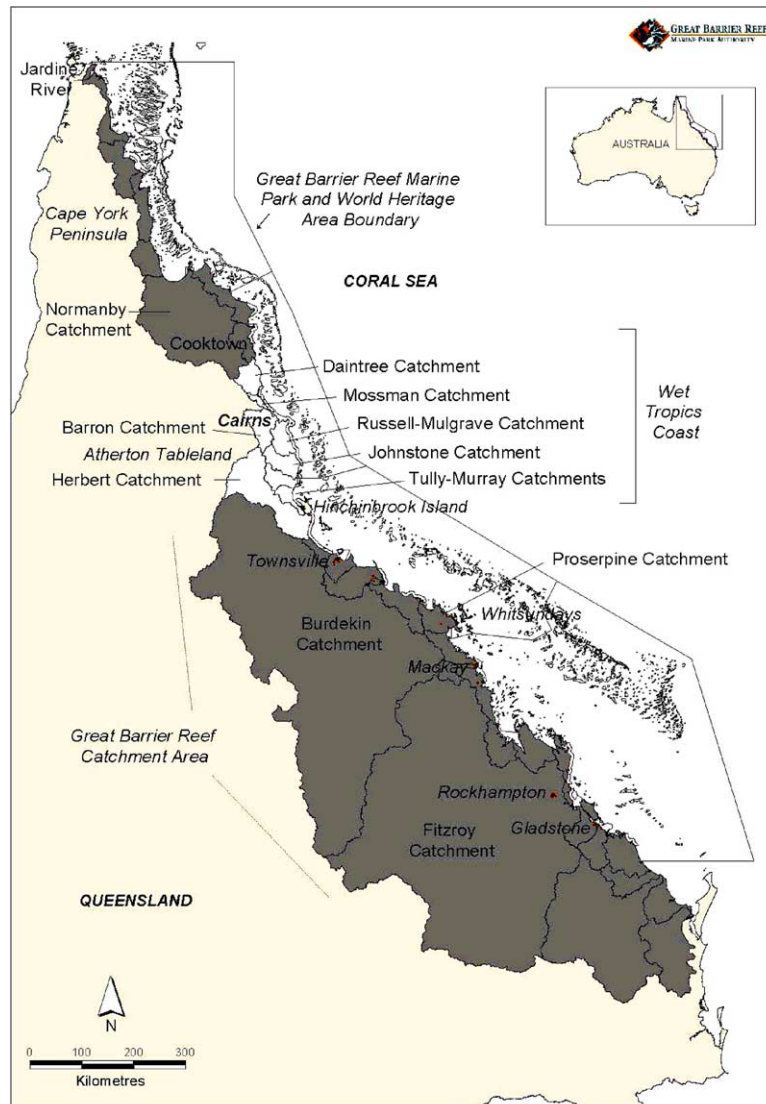


Fig. 1. Great Barrier Reef World Heritage Area and Catchment Area.

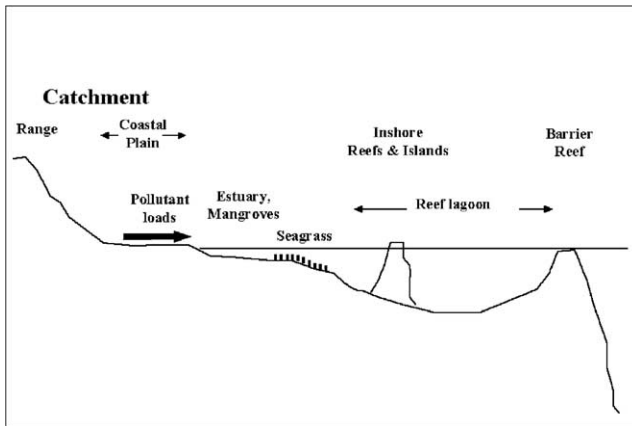


Fig. 2. Schematic cross-section of major GBR components.

Of considerable importance to the work reported here are the catchment areas adjacent to the reef (Fig. 1). Prior to European settlement these were a natural source of sediments, nutrients and a wide range of organic materials to the GBRWHA. Since European settlement (c. 1850) nearly 80% of the reef catchment has experienced changed land use. Current developed land use is estimated at grazing 77%, cropping (sugarcane) 1%, horticulture 0.2%, cotton 0.2% and urban <0.5% (Gilbert et al., 2003). This has resulted in large increases in the loads of natural pollutants, particularly sediment and nutrients (Moss et al., 1992; Neil et al., 2002; Furnas, 2003) and the appearance of man-made pollutants, particularly various herbicides. Modelling studies suggest that sediment load to the GBR has risen by a factor of four, total nitrogen by a factor of three, nitrate and total phosphorus by ten compared to pre-European conditions (Furnas, 2003; Brodie et al., 2003). Herbicides have been detected in flood waters of a number of rivers (Mitchell et al., 2004; McMahon et al., 2003) and in intertidal and subtidal sediments and seagrass along much of the GBR coast (Haynes et al., 2000a).

3. Developing water quality guidelines

A process for developing targeted water quality guidelines is outlined in Fig. 3 and is discussed below.

3.1. Identifying system stressors

System stressors are defined as inputs, processes or activities that impact on the viability of an ecosystem. Stressors can be natural (e.g. cyclones) or anthropogenic (e.g. habitat removal or pollutant input). Guidelines are developed to assist in managing anthropogenic stressors. When developing indicators and related guidelines, it is important to first define which stressors they are to address. Failure to do this can result in development of guidelines that lack specificity and are of limited use

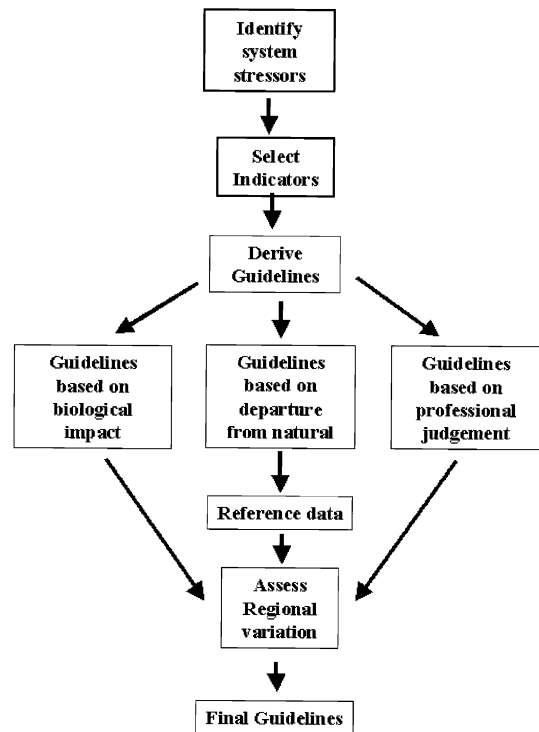


Fig. 3. An outline process for developing targeted guidelines.

for management purposes. This paper is focussed on water quality related stressors but it is noted that guidelines can equally be developed for other stressors such as fishing (e.g. catch limits) or tourism activities (e.g. limits on visitor numbers or infrastructure).

3.2. Indicator selection

Guidelines are expressed in terms of indicators. Selection of appropriate indicators is clearly of importance to ensuring that the guidelines are technically sound and are relevant to ecosystem status and management issues. Ideally indicators should:

- be linked to identified stressors,
- not be confounded by the effects of other stressors,
- not exhibit such large natural variability that the stressor signal cannot be distinguished,
- be sufficiently measured and understood to enable a useful guideline to be developed.

Types of indicators include:

- Pressure indicators—measures of the stressor itself. These can be external to the system (e.g. measures of loads of a pollutant entering a system) or they can be measures of the stressor once it enters the system. The latter are most commonly expressed as some measure of water quality.

- Condition indicators—measures of the impact of a stressor on an ecosystem or organisms in the ecosystem. These are measures of ecosystem structure and functioning: e.g. species diversity or growth rates. For management purposes, it is important to have both pressure and condition indicators, so that both cause and effect can be assessed and quantified.

Pressure indicators can usually be clearly linked to specific stressors. In contrast, more than one stressor often affects condition indicators and this creates interpretation difficulties. For example, an apparent decline in the condition of a number of coral reefs in the GBRWHA has been recorded over the past 10 years (van Woesik et al., 1999; Wachenfeld et al., 1998). However, because the condition indicators used are very general indicators of reef health, there is ongoing debate about whether these declines are due to cyclones, crown of thorns starfish, temperature bleaching or changes in water quality, or to a combination of several causes. This in turn leads to debate about what specific management action can be justified. It is important that condition indicators specific to particular stressors be developed, although the synergistic effects of different stressors make this challenging.

3.3. Deriving guidelines

There are three main approaches for deriving guideline values:

- Direct measurement of biological impacts.* Here, guideline values are based on testing the impacts of a stressor on a target organism (e.g. toxicity testing). This approach is appropriate for stressor stressors that have direct measurable impacts; e.g. toxicants, dissolved oxygen, light attenuation. It is less appropriate for stressors such as nutrients, whose threshold impacts are more complex. This approach requires information on stressor/condition relationships that can usually only be gained from controlled laboratory experiments.
- Departure from natural or reference condition.* This approach is based on the premise that small departures from natural baseline (also termed reference condition) are acceptable. It is suitable for biological condition indicators and also for indirect stressor indicators such as nutrients. This approach requires:
 - A good knowledge of the natural or reference condition based on adequate reference data sets. These data sets must be sourced from sites that are deemed to be in natural or near-natural condition. In practice, this may not always be possible so that often ‘best available’ sites have to be used.

- A value judgement on what quantum of departure from natural or reference is acceptable. This is commonly based on best technical judgement. The ANZECC, 2000 Guidelines suggest a default whereby guideline values are based on the 20th and/or 80th percentile values (whichever is appropriate) of a substantial reference data set.

- Professional judgement.* In some cases, appropriate data for a reference ecosystem do not exist, either because no appropriate reference sites exist or insufficient study has been undertaken to provide an adequate data base. In these cases professional judgement may be used, backed by appropriate scientific information, other guideline documents, and the scientific literature.

3.4. Spatial and temporal variability issues

While indicators themselves may be applicable across regional boundaries, their associated guideline values commonly are not. For example, chlorophyll-*a* concentrations in inshore waters are naturally higher than in offshore waters and also exhibit latitudinal gradients (Furnas and Brodie, 1996; Haynes et al., 2001; Brodie et al., 1997). The implication of this variability is that guideline values must include defined spatial limits. This is commonly achieved through definition of both water types (e.g. estuaries, inshore and offshore waters) and regional domains and deriving guideline values specific to each subdivision. The extent of subdivision will depend on the variability of the particular indicator, but is often a compromise between the need for specificity and the need to limit the numbers of guideline values to a practical level.

Indicator values exhibit temporal variation associated with both regular seasonal cycles and also with more random events such as floods. Seasonal variation can be addressed through season-specific guideline values although this greatly increases their complexity. Events often lead to such large, but short term, fluctuations in water quality that the application of simple guidelines during such events is inappropriate.

4. Guidelines for the GBRWHA

This section applies the principles and processes described above to derivation of guidelines for the GBRWHA.

4.1. Identifying stressors

Biological communities in the GBR are subject to a range of natural and anthropogenic stressors including cyclones, crown of thorns starfish (COTS), temperature

bleaching, fishing, tourism and pollutants from adjacent catchments (Wachenfeld et al., 1998; Brodie, 2003). This paper is focussed on water-borne pollutants and their effects on water quality and ecosystem condition.

GBR catchments south of Mossman (Fig. 1) have been extensively modified since European settlement started in the mid 1800s. It is now known that catchment development has led to large increases in pollutant loads entering rivers and subsequently being discharged into the GBRWHA (Furnas, 2003; Brodie et al., 2003). The main pollutants of concern are fine (<63 µm) sediments, nitrogen (N) phosphorus (P) and some agricultural chemicals, particularly herbicides (Brodie et al., 2001). These are the stressors addressed in this paper.

An additional stressor addressed is the effect of global warming on water temperature and coral reef bleaching. Whilst this stressor is not amenable to local management action it is nevertheless useful to have some measure of its impact thresholds.

Stressors such as fishing, habitat destruction and human recreation are outside the scope of this paper, but guidelines for these other stressors do need to be developed. The effects of fishing stress have been addressed by the recent closure of over 30% of reef areas to commercial and recreational fishing (Fernandes et al., in press).

4.2. Sediments

4.2.1. Sediments: pressure indicators and guidelines

Fine sediments cause reductions in water clarity which can impact on light-dependent organisms such as seagrasses, algae and corals. Fine sediments also affect these organisms through direct settlement, if settlement rates are too high. Potential pressure indicators would therefore include measures of clarity (light attenuation or indirectly via turbidity or suspended matter concentration) and measures of sedimentation rate.

It is difficult to develop guidelines for water clarity or sedimentation based on departure from natural. Clarity exhibits great natural variability in inshore systems, largely related to wind resuspension of fine bottom sediments (Orpin et al., 1999; Larcombe et al., 1995). Sedimentation rates are also variable and in any case there is limited data on this indicator. Equally, we have little knowledge of natural (pre 1850) levels of clarity or sedimentation rates, although coral core records may allow some assessment of these in the future (McCulloch

et al., 2003). Recent developments in remote sensing of suspended matter will allow the gathering of much larger data sets (Brando and Dekker, 2003). These data sets may allow development of more reliable guidelines, although these would have to be based on no further deterioration from current condition rather than on natural condition.

An alternative approach is to derive guidelines based on the known biological effects of fine sediments. For example, studies on the relationship between light availability and seagrass survival (Longstaff, 2003) have defined the long-term quantity of light required for survival (Table 1). Values of this nature can be used as a basis for setting guidelines for light availability or water clarity. Given that these values represent minimum requirements, and also the difficulties and errors involved in assessing light availability values at test sites, it is suggested that guideline values should be set well above minimum values. Somewhat arbitrarily, it is proposed that light availability guidelines be set at values at least 50% higher than these minimum values (values in brackets in Table 1). Application of these guidelines would involve calculating reductions of light penetration that might occur due to a particular disturbance and using the guidelines to predict resultant reductions in depth range of seagrasses.

Reduced water clarity can also cause reduction in photosynthesis and growth in corals (Rogers, 1983; Telesnicki and Goldberg, 1995), thus reducing the depth range within which corals can survive or continue active reef growth (Loya, 1976; Yentsch et al., 2002; Anthony and Fabricius, 2000). However, the situation is more complex than for seagrass, and at this stage there is no clear basis for setting guidelines. Nevertheless this is seen as an approach worth pursuing.

Fine sediments can also affect corals through smothering and abrasion caused by direct settlement (Rogers, 1990; van Katwijk et al., 1993; West and Van Woesik, 2001; McClanahan and Obura, 1997; Philipp and Fabricius, 2003). Corals use energy to remove sediment through polyp motion and mucus shedding, and this may reduce coral fitness (Riegl and Riegl, 1995). Recovery from sedimentation stress varies between species (Stafford-Smith and Ormond, 1992; Wesseling et al., 1999), and is lower for juvenile corals than for adult colonies (Hunte and Wittenberg, 1992; Fabricius et al., 2003). A further important effect of sedimentation is

Table 1
Long-term (>10 weeks) minimum light requirements for three species of seagrass

	<i>Halophila ovalis</i>	<i>Halodule pinnatifolia</i>	<i>Zostera capricorni</i>
Mol photon m ⁻² d ⁻¹	<5 (7.5) ^a	<10 (15)	10 (15)
% surface light	~6% (10%)	~20% (30%)	~30% (45%)

^a Numbers in brackets are suggested guideline values based on Longstaff's data.

the inhibition of recruitment (Tomascik and Sander, 1987; Babcock and Davies, 1991; Wittenberg and Hunte, 1992; Gilmour, 1999; Babcock and Smith, 2002).

Conservative tolerance limits of 10–30 mg dry weight sediment deposited $\text{cm}^{-2}\text{day}^{-1}$ have been proposed (Rogers, 1990; Pastorok and Bilyard, 1985; Hawker and Connell, 1989). However, such threshold values need to be tested under local circumstances before they are adopted.

4.2.2. Sediments: condition indicators and guidelines

Seagrass depth range has been used as a biological condition indicator to assess the impacts of changes in light attenuation (Abal and Dennison, 1996). Its use is based on the premise that reduced light availability will reduce the maximum depth at which seagrass can survive. For existing seagrass beds it would be possible to develop guidelines based on minimal change from existing depth ranges. Given natural variation in water clarity and hence seagrass depth range, the guidelines would have to be developed on a site-specific basis. This approach has been used with some success in Moreton Bay (Abal and Dennison, 1996). Measuring depth ranges of corals might be similarly useful, but has yet to be proven.

A recently developed indicator from coral cores, the Ba/Ca ratio (McCulloch et al., 2003) appears to be related to the amount of suspended matter in river discharge. Terrestrial sediments in flood plumes desorb Ba when they encounter increased salinity in coastal waters and this is incorporated into coral skeletons. It has been shown that Ba/Ca ratios in coral cores significantly increased at about the time of European settlement of reef catchments and it is surmised this was due to increased sediment loads in flood plumes. A guideline could be developed based on historical ratios but it would need to be tailored to local conditions since the distance of a reef from a river mouth would also affect the natural Ba/Ca ratio. Other tracers of terrestrial material in coral cores including yttrium and manganese are also currently being investigated (Kamber, personal communication; Kamber and Webb, 2001).

Biological indicators of coral reef condition that relate directly to water quality are currently under development (Fabricius et al., 2004). These include juvenile coral recruitment and survival. These indicators are not specific to sediments but rather are related to water quality stressors in general. These indicators are not yet developed to the point where guidelines can be set.

4.3. Nutrients (N&P)

4.3.1. Nutrients: pressure indicators and guidelines

Although terrestrial nutrient inputs to the reef lagoon have increased, concentrations of inorganic nutrients in the reef lagoon remain low due to rapid uptake by phy-

toplankton (Furnas and Brodie, 1996). Increased concentrations are observed only for short periods following major flood events (Devlin et al., 2001; Devlin and Brodie, 2004); upwelling events (Furnas and Mitchell, 1996); and cyclonic resuspension events (Furnas, 1989). Inorganic nutrient concentrations on their own may not be a particularly good indicator of nutrient stress to the reef. Nevertheless, nutrient guidelines are a useful benchmark and can be applied to localised management purposes, e.g. around discharges or in relation to intensive developments. Since inorganic nutrients are quickly taken up by phytoplankton, the effects of increased nutrient loads may be expressed as increased phytoplankton biomass, which is readily measured as chlorophyll-*a* concentration (Brodie and Furnas, 1994).

There are reasonably extensive data sets for both nutrient and chlorophyll-*a* concentrations in both estuarine and coastal waters of the GBRWHA. From these data it is possible to set guidelines based on the ANZECC (2000) Guidelines methodology of setting guideline values at the 20th or 80th percentile (whichever is appropriate for the indicator) of the range of values at undisturbed systems.

Inshore areas south of about Cooktown (16°S) are all impacted to some degree by land run-off and therefore contain no true reference (unmodified) sites. Therefore, guidelines developed for both nutrients and chlorophyll-*a* in these areas need to have a caveat associated with the guideline, in that they may not reflect true natural condition.

To account for regional variation, the GBRWHA has been divided into three latitudinal regions: (a) Cape York (Cape York to Cape Flattery), (b) the Wet Tropics (Cape Flattery to Hinchinbrook Island) and (c) the Dry Tropics (Hinchinbrook Island to Gladstone). Within these regions four water types are identified—estuaries, enclosed coastal embayments (shallow inshore waters, bays, channels), inshore waters (<15 km offshore) and offshore waters (>15 km offshore).

Data on the concentrations of nutrients and chlorophyll in GBR waters has been collected by the Australian Institute of Marine Science, the Great Barrier Reef Marine Park Authority and the Queensland Environmental Protection Agency. The data set is being continuously updated, but summaries and analysis have been published through time, e.g. Furnas and Brodie (1996), Brodie and Furnas (1996), Brodie et al. (1997), Devlin et al. (2001), Haynes et al. (2001), Furnas and Mitchell (1997), Furnas (2003) and GBR long-term chlorophyll monitoring program (2004). The chlorophyll database collected between 1991 and the present has 650 records from the Cape York area, 800 records from the Wet Tropics area and 950 records from the Dry Tropics area (GBR long-term chlorophyll monitoring program, 2004). The nutrient database (AIMS Biological Oceanography Group) contains over 4000

records from the period 1978–2004 from throughout the Great Barrier Reef. EPA datasets for estuaries are based on monthly sampling in a number of wet and dry tropics estuaries over 4 or more years. Based on the 80th percentiles of these data sets, the values in Table 2 are proposed as guideline values for the regions and water types identified above.

Studies on the direct biological effects of elevated nutrients on corals and tridacnid clams have been in progress in the GBR for the last decade and follow up the pioneering work of Kinsey and Davies (1979) and Kinsey and Domm (1974) at One Tree Island. Work has occurred at Orpheus Island Research Station using flow-through aquarium systems (Rasmussen and Cuff, 1990; Belda et al., 1993), the ENCORE project at One Tree Island Research Station, using micro-atolls as research units (Ward and Harrison, 2000; Koop et al., 2001), and worldwide studies on the effects of nutrients on corals (Ferrier-Pages et al., 2001).

The direct effects of dissolved inorganic nutrients, such as nitrate, ammonium and phosphate, on adult corals are complex (Szmant, 2002; Fabricius, 2004). Studies investigating the effects of elevated nutrients on coral physiology and health have yielded variable responses (Tomascik and Sander, 1985; Koop et al., 1999). In some experiments increased nutrient concentrations increased coral growth, but reduced skeletal density, while in others high concentrations had the opposite ef-

fect. High nutrient concentrations appear to increase the density of zooxanthellae in the coral and hence change the balance of energy, CO₂ and nutrients between zooxanthellae and coral (Muscatine et al., 1989; Marubini and Davies, 1996; Ferrier-Pages et al., 2001). Even small increases in the concentrations of nutrients appear to have detrimental effects on the reproduction of corals (Ward and Harrison, 2000; Harrison and Ward, 2001) with, for example, decreased larval production in slightly elevated ammonium concentrations (Cox and Ward, 2002). The variability in response to different nutrient levels means that direct effects guidelines for nutrients cannot be derived at this stage.

The indirect effects of nutrients on coral reef systems are also complex but becoming clearer (Fabricius et al., 2004; Fabricius, 2004). Many indicators of reef condition, such as hard and soft coral diversity, reef growth, bioerosion, coralline algal abundance, coral recruitment and recruit survivorship, can be correlated with water quality gradients (van Woesik et al., 1999; Fabricius and De'ath, 2001a,b; Fabricius and De'ath, 2004; Fabricius et al., 2004; West and Van Woesik, 2001). The actual nature of the interaction causing reef degradation is complex and still poorly understood. Two of the better understood interactions are those involving the formation of muddy marine snow and subsequent coral mortality through smothering (Fabricius and Wolanski, 2000; Wolanski and Spagnol, 2000; Wolanski et al., 2003; Fabricius et al., 2003) and interactions between macroalgal growth, grazing pressure on the algae, nutrient enhancement of algal growth and hence coral-algal competition for space on coral reefs (McCook, 1999; McCook et al., 2001; Miller, 1998; Lapointe, 1997; Hughes et al., 1999; Stimson and Larned, 2001). While correlations of nutrient and chlorophyll concentrations with reef condition are emerging from these studies it is still too early to derive guidelines.

Another important nutrient-related interaction on reefs of the GBR, and through the Indo-Pacific generally, is that between the coral-eating crown of thorns starfish (*Acanthaster planci*) and reef condition. It is now believed that outbreaks of *A. planci* are associated with broad scale nutrient enrichment from land runoff and subsequent phytoplankton blooms leading to enhanced survivorship of *A. planci* larvae (Brodie et al., 2004). The critical chlorophyll concentration range at which larval survivorship becomes significantly enhanced is 0.5–0.8 µg l⁻¹ (Brodie et al., 2004). It is thus possible to use a chlorophyll concentration of 0.5 µg l⁻¹ in the larval period of *A. planci* (November to February) as a threshold guideline to ensure *A. planci* outbreaks are minimised.

4.3.2. Nutrients: condition indicators and guidelines

A simple biomarker that has been used to assess human-induced nitrogen enrichment is δ¹⁵N the ratio

Table 2
Nutrient and chlorophyll-*a* guidelines

Indicator	Estuary	Enclosed coastal	Inshore	Offshore
<i>Cape York region</i>				
DIN µg l ⁻¹	nd	nd	2	1
TDN µg l ⁻¹	nd	nd	135	110
PN µg l ⁻¹	nd	nd	25	25
TN µg l ⁻¹	nd	nd	160	135
TP µg l ⁻¹	nd	nd	30	30
FRP µg l ⁻¹	nd	nd	3	3
Chla µg l ⁻¹	nd	nd	0.5	0.3
<i>Wet tropics region</i>				
DIN µg l ⁻¹	45	25	3	4
TDN µg l ⁻¹	nd	nd	120	110
PN µg l ⁻¹	nd	nd	25	20
TN µg l ⁻¹	250	160	145	130
TP µg l ⁻¹	20	20	20	10
FRP µg l ⁻¹	5	5	3	4
Chla µg l ⁻¹	3	2	0.6	0.3
<i>Dry tropics region</i>				
DIN µg l ⁻¹	20	10	7	3
TDN µg l ⁻¹	nd	nd	130	100
PN µg l ⁻¹	nd	nd	25	20
TN µg l ⁻¹	300	200	155	120
TP µg l ⁻¹	25	20	20	12
FRP µg l ⁻¹	8	6	6	5
Chla µg l ⁻¹	4	2	0.6	0.5

DIN—dissolved inorganic N, TDN—total filterable N (0.45 µm), PN—particulate N, TN—total N, TP—total P, FRP—filterable reactive P, nd—no data on which to base a guideline.

of ^{15}N to ^{14}N . The $\delta^{15}\text{N}$ value in the tissues of macroalgae, seagrass and mangroves has been shown to reflect the presence of N from N rich point discharges such as from prawn farms and sewage treatment plants (Jones et al., 2001; Costanzo et al., 2001), but can also reflect more general catchment related disturbances of the N cycle (Udy and Bunn, 2001). In near coastal systems, background values of $\delta^{15}\text{N}$ are generally around 3–4‰, while disturbed sites can be up to 15‰. On the GBR, values of $\delta^{15}\text{N}$ in the coral *Porites lobata* range from 5.0‰ to 5.5‰ inshore, low values on the mid-shelf (3.8‰) and higher values offshore (5.2‰) (Sammarco et al., 1999). While further work is required, available data suggests that an appropriate $\delta^{15}\text{N}$ guideline would be around 6‰. As noted for sediments, biological indicators of coral reef condition that relate directly to water quality are currently under development (Fabricius et al., 2004).

4.4. Toxicants

4.4.1. Toxicants: pressure indicators and guidelines

The ANZECC (2000) Guidelines provide values for a wide range of pesticides, organic compounds and heavy metals in waters and sediments. In the absence of other published guidelines, these are the best defaults available. However, it is still desirable to develop more reef specific guidelines.

A large body of literature is available (Hutchings and Haynes, 2000), which details both the natural concentrations of toxicants and also some species-specific toxicant relationships in the GBR region. However, most authors rely on the ANZECC (2000) Guidelines values for assessing their data and few have taken the additional step of using their data to derive more locally relevant guideline numbers.

The large amount of toxicant data scattered throughout the literature could contribute to guidelines in two ways:

- By establishing baseline or reference condition for pollutants that occur naturally at low levels (e.g. metals and some hydrocarbons) in waters, sediments and a wide range of biota. While such reference values do not give direct information on toxicity, knowledge of reference levels of, for example, metals in shellfish, is a very useful tool in assessing if metal contamination has occurred. Reference values for water and sediment are also a useful reality check for the ANZECC (2000) guideline values.
- To derive direct impact guidelines based on the toxicity relationships for pollutants. The advent of Phyto PAM technology (Jones et al., 1999) has for example provided a very powerful tool to assess sublethal effects of herbicides on organisms that are reliant wholly or in part on photosynthesis. A number of studies (Haynes et al., 2000b) have used this technol-

ogy to assess the threshold levels at which herbicides that begin to impact on seagrasses and also corals. This type of data can be used to derive impact based guideline values.

It is not possible in this paper to review all the literature on the natural toxicant levels and the direct impacts of toxicants, but it is suggested that such a review would be fruitful in developing guidelines specific to the GBRWHA.

However, there are sufficient readily available data to develop a preliminary guideline for the herbicide diuron, which is a common pollutant entering the GBR system. Data on toxicity to mangroves (Duke et al., 2003; Duke and Bell, 2004), seagrass (Haynes et al., 2000b) and corals (Jones et al., 2003) are available.

Diuron suppresses photosynthesis in the seagrasses *Cymodocea serrulata*, *Halophila ovalis* and *Zostera capricorni* at concentrations in seawater of $10\mu\text{g l}^{-1}$, $0.1\mu\text{g l}^{-1}$ and $0.1\mu\text{g l}^{-1}$ respectively, and no full recovery occurred after five days of recovery (no applied diuron) period (Ralph, 2000; Haynes et al., 2000b). In the corals *Acropora formosa*, *Montipora digitata* and *Seriatopora hystrix* suppression of photosynthesis occurred at an exposure of $1\mu\text{g l}^{-1}$ diuron for four days, but apparent full recovery occurred after a recovery period of four days with no diuron. Diuron at $10\mu\text{g l}^{-1}$, slightly bleached corals after four days exposure and there was incomplete recovery after a four day recovery period (Jones et al., 2003). Long-term effects (reduction in health indicators) were observed in the mangroves *Rhizophora stylosa* and *Avicennia marina* after 16 days exposure to 4mg kg^{-1} of diuron in sediments (Duke et al., 2003). In field observations, *A. marina* dieback was observed near Mackay where diuron concentrations in sediments were in the range $6\text{--}8\mu\text{g l}^{-1}$ (Duke et al., 2003; Duke and Bell, 2004).

These results suggest that diuron can have direct effects on corals and seagrass at concentrations as low as $0.1\mu\text{g l}^{-1}$ and long-term effects (no full recovery) at $10\mu\text{g l}^{-1}$. A preliminary guideline for seawater should be set at $0.1\mu\text{g l}^{-1}$, i.e. the concentration above which non-reversible damage occurs.

4.5. Temperature

4.5.1. Temperature: pressure indicators and guidelines

A number of coral bleaching events related to water temperature have occurred on the GBR, the most serious and widespread in 1998 (Berkelmans and Oliver, 1999) and 2002 (Berkelmans et al., 2004). The best currently available climate change prediction models suggest bleaching will become a frequent occurrence on the GBR with severe implications for the condition of coral reefs (Hoegh-Guldberg, 1999). Analysis of satellite-derived sea surface temperature (SST) records from

Table 3

Comparison of guidelines developed in three studies

	Hawker and Connell (1989)	ANZECC (2000)		Present study	
		Inshore	Offshore	Inshore	Offshore
Chlorophyll- <i>a</i> ($\mu\text{g l}^{-1}$)	0.59	0.7–1.4	0.5–0.9	0.5–0.6	0.3–0.5
Phosphate (FRP) ($\mu\text{g l}^{-1}$)	0.25	5	2–5	3–8	3–4
Ammonium ($\mu\text{g l}^{-1}$)	0.65	1–10	1–6		
Nitrate + nitrite ($\mu\text{g l}^{-1}$)	1.31	2–8	1–4		
DIN ^a ($\mu\text{g l}^{-1}$)	2	3–18	2–10	1–5	1
TP ($\mu\text{g l}^{-1}$)		15	10	15–20	12–20
TN ($\mu\text{g l}^{-1}$)		100	100	145–155	120–135

^a DIN by addition of ammonium and nitrate + nitrite for Hawker and Connell (1989) and ANZECC.

the GBR shows that the best predictor of bleaching is maximum SST occurring over any 3-day period (Berkelmans, 2002; Berkelmans et al., 2004). Short periods (3–6 days) of high temperatures (i.e. temperatures greater than one degree Celsius above long-term mean maximums) are highly stressful to corals and result in bleaching. However bleaching can also be predicted using SST HotSpots analysis (NOAA, 2004).

HotSpots

= SST analysis

– SST interpolated maximum monthly climatology

where the SST analysis is obtained from current satellite-derived temperatures, and the SST interpolated maximum monthly climatology is obtained from historic data (NOAA, 2004).

Where HotSpots are greater than one degree Celsius, coral bleaching events are predicted. While the 3-day maximum method (Berkelmans et al., 2004) provides better predictions than anomaly-based SST analysis a full method using this techniques is not yet available. As a preliminary guideline the HotSpot method is recommended.

Increased temperatures initially impact corals by causing the expulsion of zooxanthellae. The direct impacts of thermal stress on corals could therefore be assessed using PhytoPam technology. Further development of the relationship between photosynthetic functionality and increasing temperature stress would allow development of a sublethal guideline for temperature increase, i.e. a guideline on what increases in temperature can occur without resulting in full bleaching.

4.5.2. Temperature: condition indicators and guidelines

The loss of photosynthetic functionality, assessed using PhytoPam technology, would be a good biological condition measure of thermal stress. It would be theoretically possible to develop guidelines based on this measure, but the technology is not easy to apply broadly in the field. The application of pressure guidelines using readily available satellite data appears to be the best way to assess the impacts of this particular stressor.

4.5.3. Comparison of guideline values derived in this paper with earlier guidelines

Initial work to develop nutrient and chlorophyll-*a* guidelines for the GBR and other coral reefs systems was carried out by Hawker and Connell (1989) and Bell et al. (1989). Data was drawn from the worldwide literature because little information was available for the GBR at that time.

In the late 1990s a revised set of water quality guidelines for Australian and New Zealand waters was developed (ANZECC, 2000). These guidelines included a subset for tropical waters representing trigger values for slightly disturbed systems (Tables 3.3.4 and 3.3.5 in ANZECC, 2000).

Comparing the guidelines developed in this study with these earlier guideline sets where there are parameters in common is instructive (see Table 3). The two sets of guidelines were similar for chlorophyll-*a*. For phosphate, however the Hawker and Connell guideline is an order of magnitude lower than the values derived in this paper, most likely due to Hawker and Connell using Caribbean data to determine their guideline. Caribbean waters generally have lower concentrations of phosphate than waters adjacent to reefs in the Indo-Pacific (Kleypus et al., 1999). The guideline values for the other parameters, DIN, TP and TN are relatively similar, although we now realise more clearly from decades of monitoring that ambient DIN concentrations in GBR waters are very low, while dissolved organic nitrogen (DON) concentrations are high and relatively consistent. DON makes up the major proportion of the TN in GBR waters (Furnas and Brodie, 1996).

5. Conclusions

This paper examines water quality related pressure and condition indicators for the Great Barrier Reef World Heritage Area. The relevance of these indicators with respect to regional ecosystem health and the availability of data on which to base guideline values is discussed.

The most extensive data are generally available for indicators of stressor concentration (i.e. pressure indicators). Guideline values for these indicators can be developed based on some arbitrary small departure from natural or baseline condition, although large natural variations in baseline values often complicate this. Such guidelines may meet the needs of the precautionary principle but their relationship to biological impact is not well defined and hence they are best applied in combination with other types of indicators.

More reliable guidelines can be developed from well researched direct biological impact relationships. This approach works well for many toxicants but the data for this region is fragmented and needs to be comprehensively compiled. For some indicators, e.g. nutrients, impact relationships are complex and highly equivocal.

Indicators of biological condition response to stressors are important in that they tell us how a system is actually coping with stress. However, the development of such indicators for the GBR region is in its infancy and there is consequently little data on which to base guidelines. Response indicators have the additional problem that they often respond to more than one stressor.

A mix of guidelines based on all types of indicators best serves management. This allows both pressure and biological condition to be independently assessed leading to a more balanced assessment of issues and required management responses.

Based on this review the following priorities are proposed for future research and data gathering:

- Establish in more detail the existing ranges of suspended particulate matter and chlorophyll-*a* in different parts of the reef lagoon based on remote sensing data.
- Establish light requirements of a range of key coral reef species.
- Establish tolerances of corals to sediment deposition.
- Establish baseline $\delta^{15}\text{N}$ values for coral reefs in more detail.
- Develop reef condition indicators that can be related directly to water quality impacts.
- Assess potential of isotopic biomarkers such as the Ba/Ca ratio for use as indicators of reef condition.

There are a number of ecosystem components of the GBRWHA for which we currently have no useful indicators of biological condition. These include the extensive sandy and muddy sediment areas of the reef lagoon. At this stage we have very little knowledge of how or if increased pollutant loads from the land affect these system components. These components are likely to be less sensitive to catchment sourced stressors than coral reefs but they nevertheless merit some attention.

A question of great relevance to governments and the public is whether increased terrestrial pollutant loads are impacting GBRWHA ecosystems, and in particular the coral reefs. Since we do not yet have adequate indicators to determine the impacts on reefs from water quality (as distinct from other types of impacts), it seems clear that developing such indicators, obtaining adequate data sets and developing soundly based guidelines must be a high priority.

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